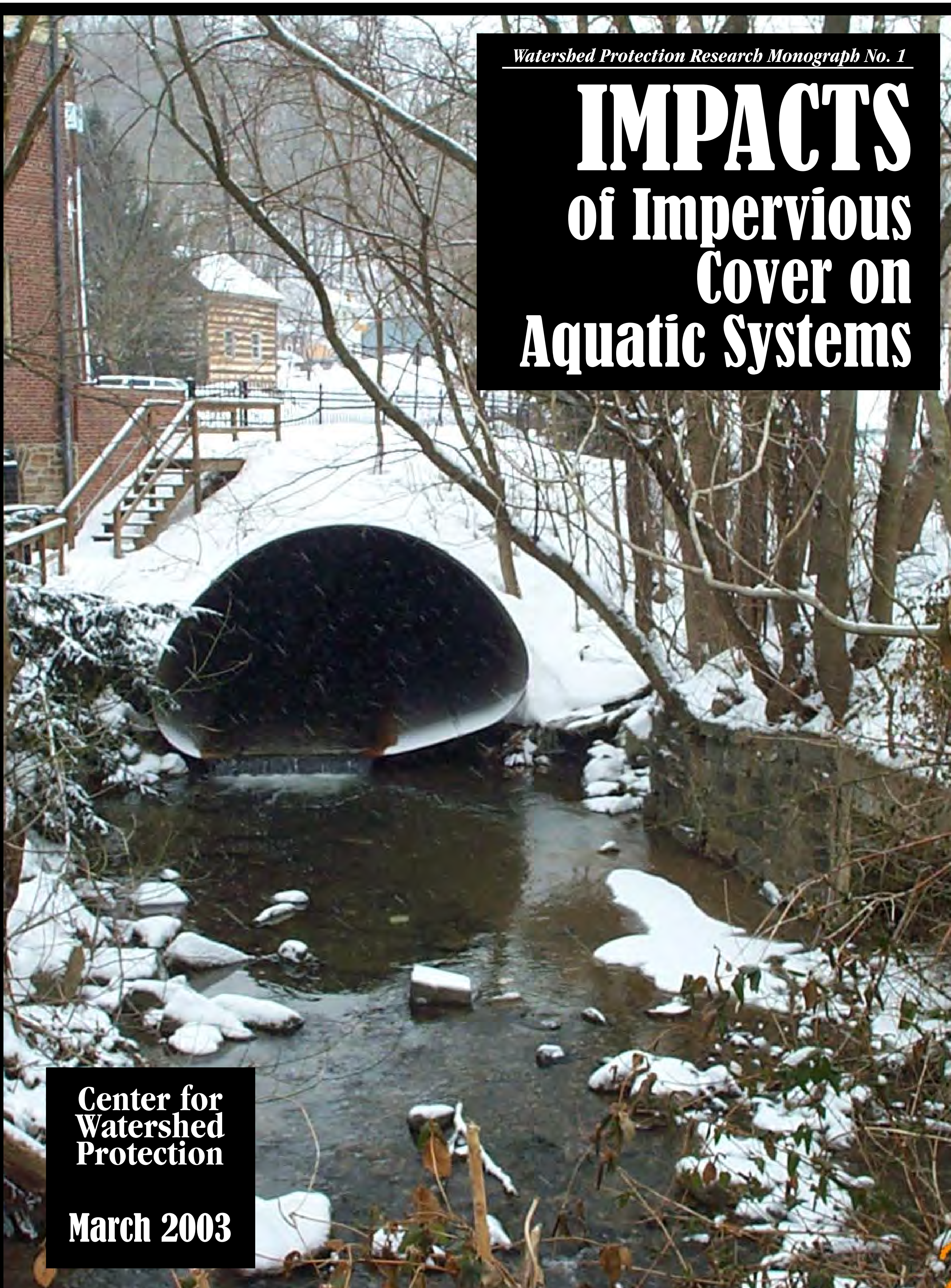
ATTACHMENT 3

Watershed Protection Research Monograph No. 1

IMPACTS of Impervious Cover on Aquatic Systems



Center for Watershed Protection

March 2003

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Impacts of Impervious Cover on Aquatic Systems

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Chapter 1: Introduction

This research monograph comprehensively reviews the available scientific data on the impacts of urbanization on small streams and receiving waters. These impacts are generally classified according to one of four broad categories: changes in hydrologic, physical, water quality or biological indicators. More than 225 research studies have documented the adverse impact of urbanization on one or more of these key indicators. In general, most research has focused on smaller watersheds, with drainage areas ranging from a few hundred acres up to ten square miles.

Streams vs. Downstream Receiving Waters

Urban watershed research has traditionally pursued two core themes. One theme has evaluated the direct impact of urbanization on small streams, whereas the second theme has explored the more indirect impact of urbanization on downstream receiving waters, such as rivers, lakes, reservoirs, estuaries and coastal areas. This report is organized to profile recent research progress in both thematic areas and to discuss the implications each poses for urban watershed managers.

When evaluating the direct impact of urbanization on streams, researchers have emphasized hydrologic, physical and biological indicators to define urban stream quality. In recent years, impervious cover (IC) has emerged as a key paradigm to explain and sometimes predict how severely these stream quality indicators change in response to different levels of watershed development. The Center for Watershed Protection has integrated these research findings into a general watershed planning model, known as the impervious cover model (ICM). The ICM predicts that most stream quality indicators decline when watershed IC exceeds 10%, with severe degradation expected beyond 25% IC. In the first part of this review, we critically analyze the scientific basis for the ICM and explore some of its more interesting technical implications.

While many researchers have monitored the quality of stormwater runoff from small watersheds, few have directly linked these pollutants to specific water quality problems within streams (e.g., toxicity, biofouling, eutrophication). Instead, the prevailing view is that stormwater pollutants are a downstream export. That is, they primarily influence downstream receiving water quality. Therefore, researchers have focused on how to estimate stormwater pollutant loads and then determine the water quality response of the rivers, lakes and estuaries that receive them. To be sure, there is an increasing recognition that runoff volume can influence physical and biological indicators within some receiving waters, but only a handful of studies have explored this area. In the second part of this review, we review the impacts of urbanization on downstream receiving waters, primarily from the standpoint of stormwater quality. We also evaluate whether the ICM can be extended to predict water quality in rivers, lakes and estuaries.

This chapter is organized as follows:

1.1 A Review of Recent Urban Stream Research and the ICM
1.2 Impacts of Urbanization on Downstream Receiving Waters
1.3 Implications of the ICM for Watershed Managers

1.1 A Review of Recent Urban Stream Research and the ICM

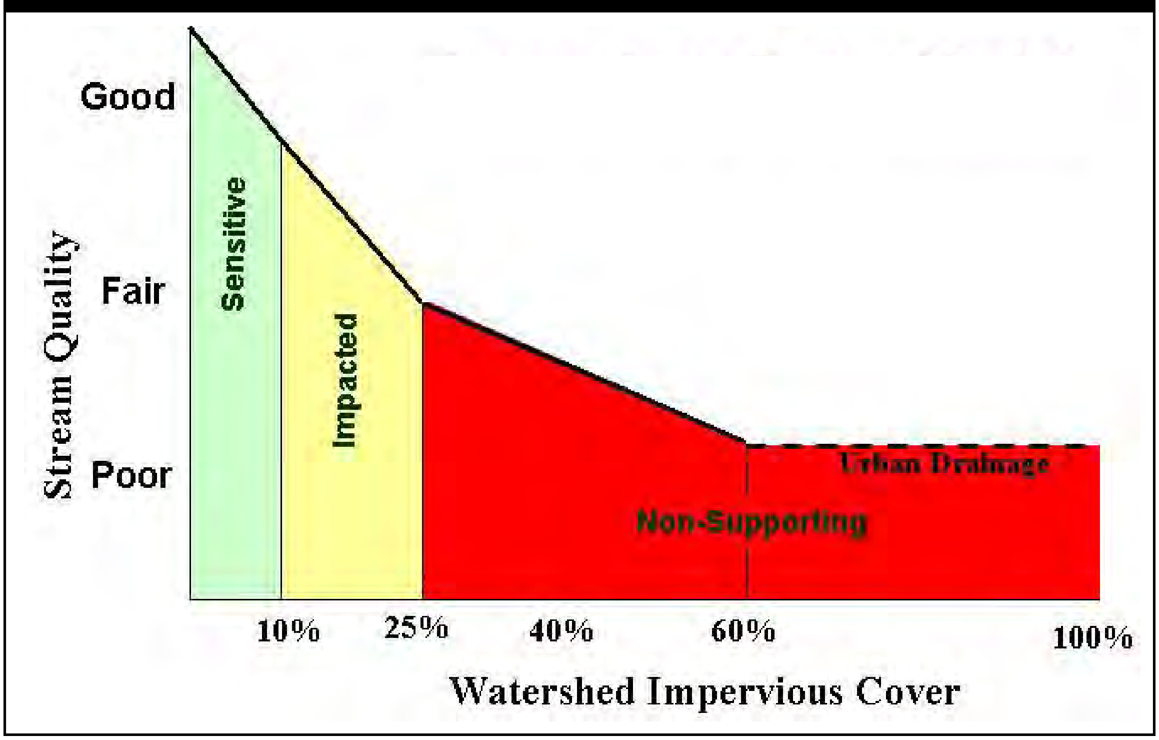
In 1994, the Center published "The Importance of Imperviousness," which outlined the scientific evidence for the relationship between IC and stream quality. At that time, about two dozen research studies documented a reasonably strong relationship between watershed IC and various indicators of stream quality. The research findings were subsequently integrated into the ICM (Schueler, 1994a and CWP, 1998). A brief summary of the basic assumptions of the ICM can be found in Figure 1. The ICM has had a major influence in watershed planning, stream classification and land use regulation in many communities. The ICM is a deceptively simple model that raises extremely complex and profound policy implications for watershed managers.

The ICM has been widely applied in many urban watershed settings for the purposes of small watershed planning, stream classification, and supporting restrictive development regulations and watershed zoning. As such, the ICM has stimulated intense debate among the planning, engineering and scientific communities. This debate is likely to soon spill over into the realm of politics and the courtroom, given its potential implications for local land use and environmental regulation. It is no wonder that the specter of scientific uncertainty is frequently invoked in the ICM debate, given the land use policy issues at stake. In this light, it is helpful to review the current strength of the evidence for and against the ICM.

The ICM is based on the following assumptions and caveats:

- Applies only to 1st, 2nd and 3rd order streams.
- Requires accurate estimates of percent IC, which is defined as the total amount of impervious cover over a subwatershed area.
- Predicts <u>potential</u> rather than <u>actual</u> stream quality. It can and should be expected that some streams will depart from the predictions of the model. For example, monitoring indicators may reveal poor water quality in a stream classified as "sensitive" or a surprisingly high biological diversity

Figure 1: Impervious Cover Model



score in a "non-supporting" one. Consequently, while IC can be used to initially diagnose stream quality, supplemental field monitoring is recommended to actually confirm it.

- Does not predict the precise score of an individual stream quality indicator but rather predicts the average behavior of a group of indicators over a range of IC. Extreme care should be exercised if the ICM is used to predict the fate of individual species (e.g., trout, salmon, mussels).

- "Thresholds" defined as 10 and 25% IC are not sharp "breakpoints," but instead reflect the expected transition of a composite of individual indicators in that range of IC. Thus, it is virtually impossible to distinguish real differences in stream quality indicators within a few percentage points of watershed IC (e.g., 9.9 vs. 10.1%).

- Should only be applied within the ecoregions where it has been tested, including the mid-Atlantic, Northeast, Southeast, Upper Midwest, and Pacific Northwest.

- Has not yet been validated for non-stream conditions (e.g., lakes, reservoirs, aquifers and estuaries).

- Does not currently predict the impact of watershed treatment.

In this section, we review available stream research to answer four questions about the ICM:

1. Does recent stream research still support the basic ICM?
2. What, if any, modifications need to be made to the ICM?
3. To what extent can watershed practices shift the predictions of the ICM?
4. What additional research is needed to test the ICM?

1.1.1 Strength of the Evidence for the ICM

Many researchers have investigated the IC/stream quality relationship in recent years. The Center recently undertook a comprehensive analysis of the literature to assess the scientific basis for the ICM. As of the end of 2002, we discovered more than 225 research studies that measured 26 different urban stream indicators within many regions of North America. We classified the research studies into three basic groups.

The first and most important group consists of studies that directly test the IC/stream quality indicator relationship by monitoring a large population of small watersheds. The second and largest group encompasses secondary studies that indirectly support the ICM by showing significant differences in stream quality indicators between urban and non-urban watersheds. The third and last group of studies includes widely accepted engineering models that explicitly use IC to directly predict stream quality indicators. Examples include engineering models that predict peak discharge or stormwater pollutant loads as a direct function of IC. In most cases, these relationships were derived from prior empirical research.

Table 1 provides a condensed summary of recent urban stream research, which shows the impressive growth in our understanding of urban streams and the watershed factors that influence them. A negative relationship between watershed development and nearly all of the 26 stream quality indicators has been established over many regions and scientific disciplines. About 50 primary studies have tested the IC/stream quality indicator relationship, with the largest number looking at biological indicators of stream health, such as the diversity of aquatic insects or fish. Another 150 or so secondary studies provide evidence that stream quality indicators are significantly different between urban and non-urban watersheds, which lends at least indirect support for the ICM and suggests that additional research to directly test the IC/stream quality indicator

Table 1: The Strength of Evidence: A Review of the Current Research on Urban Stream Indicators						
Stream Quality Indicator	**#**	**IC**	**UN**	**EM**	**RV**	**Notes**
Increased Runoff Volume	2	Y	Y	Y	N	extensive national data
Increased Peak Discharge	7	Y	Y	Y	Y	type of drainage system key
Increased Frequency of Bankfull Flow	2	?	Y	N	N	hard to measure
Diminished Baseflow	8	?	Y	N	Y	inconclusive data
Stream Channel Enlargement	8	Y	Y	N	Y	stream type important
Increased Channel Modification	4	Y	Y	N	?	stream enclosure
Loss of Riparian Continuity	4	Y	Y	N	?	can be affected by buffer
Reduced Large Woody Debris	4	Y	Y	N	?	Pacific NW studies
Decline in Stream Habitat Quality	11	Y	Y	N	?	
Changes in Pool Riffle/Structure	4	Y	Y	N	?	
Reduced Channel Sinuosity	1	?	Y	N	?	straighter channels
Decline in Streambed Quality	2	Y	Y	N	?	embeddedness
Increased Stream Temperature	5	Y	Y	N	?	buffers and ponds also a factor
Increased Road Crossings	3	?	Y	N	?	create fish barriers
Increased Nutrient Load	30+	?	Y	Y	N	higher stormwater EMCs
Increased Sediment Load	30+	?	Y	N	Y	higher EMCs in arid regions
Increased Metals & Hydrocarbons	20+	?	Y	Y	N	related to traffic/VMT
Increased Pesticide Levels	7	?	Y	N	Y	may be related to turf cover
Increased Chloride Levels	5	?	Y	N	Y	related to road density
Violations of Bacteria Standards	9	Y	Y	N	Y	indirect association
Decline in Aquatic Insect Diversity	33	Y	Y	N	N	IBI and EPT
Decline in Fish Diversity	19	Y	Y	N	N	regional IBI differences
Loss of Coldwater Fish Species	6	Y	Y	N	N	trout and salmon
Reduced Fish Spawning	3	Y	Y	N	?	
Decline in Wetland Plant Diversity	2	N	Y	N	?	water level fluctuation
Decline in Amphibian Community	5	Y	Y	N	?	few studies

***#**: total number of all studies that evaluated the indicator for urban watersheds*
***IC**: does balance of studies indicate a progressive change in the indicator as IC increases? Answers: Yes, No or No data (?)*
***UN**: If the answer to IC is no, does the balance of the studies show a change in the indicator from non-urban to urban watersheds? Yes or No*
***EM** Is the IC/stream quality indicator relationship implicitly assumed within the framework of widely accepted engineering models? Yes, No or No models yet exist (?)*
***RV**: If the relationship has been tested in more than one eco-region, does it generally show major differences between ecoregions? Answers: Yes, No, or insufficient data (?)*

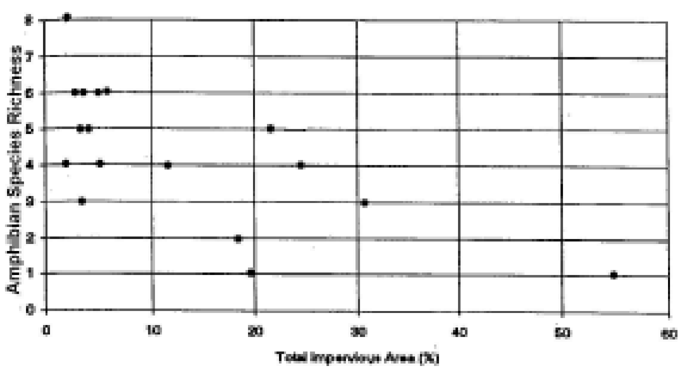
relationship is warranted. In some cases, the IC/stream quality indicator relationship is considered so strongly established by historical research that it has been directly incorporated into accepted engineering models. This has been particularly true for hydrological and water quality indicators.

1.1.2 Reinterpretation of the ICM

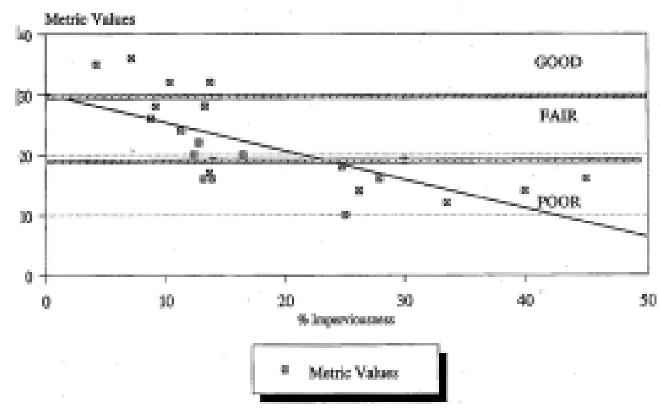
Although the balance of recent stream research generally supports the ICM, it also offers several important insights for interpreting and applying the ICM, which are discussed next.

Statistical Variability
Scatter is a common characteristic of most IC/stream quality indicator relationships. In most cases, the overall trend for the indicator is down, but considerable variation exists along the trend line. Often, linear regression equations between IC and individual stream quality indicators produce relatively modest correlation coefficients (reported r^2 of 0.3 to 0.7 are often considered quite strong).

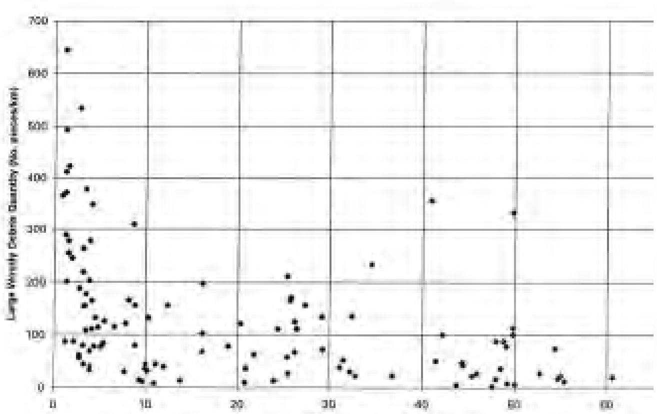
Figure 2 shows typical examples of the IC/stream quality indicator relationship that illustrate the pattern of statistical variability. Variation is always encountered when dealing with urban stream data (particularly so for biological indicators), but several patterns exist that have important implications for watershed managers.



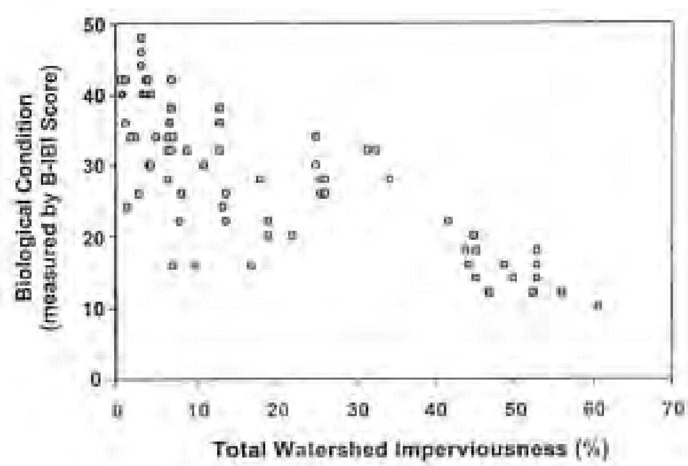
a. Fish IBI vs. IC in Fairfax, VA (Fairfax County, 2001)



b. CPSS vs. IC in Montgomery County, MD (MNCPPC, 2000)



c. Large Woody Debris vs. IC (Booth *et al.*, 1997)



d. Biological Condition vs. Total Watershed IC (Booth, 2000)

Figure 2: Typical Scatter Found in IC/Stream Quality Indicator Research

The first pattern to note is that the greatest scatter in stream quality indicator scores is frequently seen in the range of one to 10% IC. These streams, which are classified as "sensitive" according to the ICM, often exhibit low, moderate or high stream quality indicator scores, as shown in Figure 2. The key interpretation is that sensitive streams have the potential to attain high stream quality indicator scores, but may not always realize this potential.

Quite simply, the influence of IC in the one to 10% range is relatively weak compared to other potential watershed factors, such as percent forest cover, riparian continuity, historical land use, soils, agriculture, acid mine drainage or a host of other stressors. Consequently, watershed managers should never rely on IC alone to classify and manage streams in watersheds with less than 10% IC. Rather, they should evaluate a range of supplemental watershed variables to measure or predict actual stream quality within these lightly developed watersheds.

The second important pattern is that variability in stream quality indicator data is usually dampened when IC exceeds 10%, which presumably reflects the stronger influence of stormwater runoff on stream quality indicators. In particular, the chance that a stream quality indicator will attain a high quality score is sharply diminished at higher IC levels. This trend becomes pronounced within the 10 to 25% IC range and almost inevitable when watershed IC exceeds 25%. Once again, this pattern suggests that IC is a more robust and reliable indicator of overall stream quality beyond the 10% IC threshold.

Other Watershed Variables and the ICM

Several other watershed variables can potentially be included in the ICM. They include forest cover, riparian forest continuity and turf cover.

Forest cover (FC) is clearly the main rival to IC as a useful predictor of stream quality in urban watersheds, at least for humid regions of North America. In some regions, FC is simply the reciprocal of IC. For example, Horner and May (1999) have demonstrated a strong interrelationship between IC and FC for subwatersheds in the Puget Sound region (Figure 3). In other regions, however, "pre-

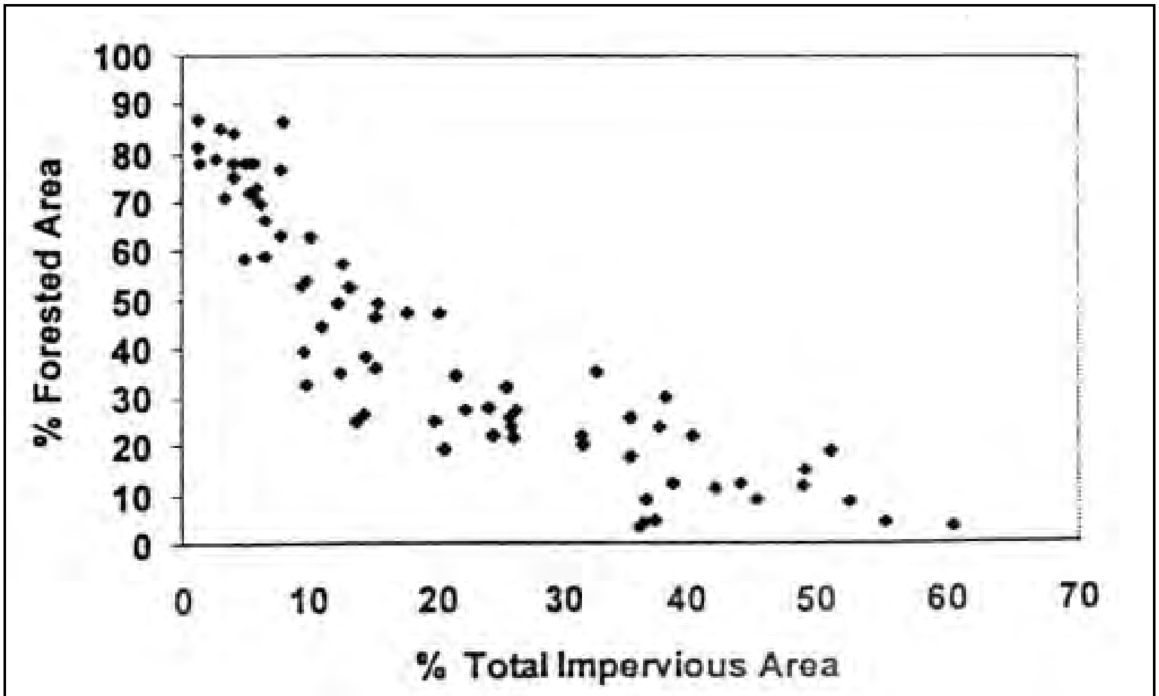


Figure 3: Relationship of IC and FC in Puget Sound Subwatersheds (Horner and May, 1999)

development" land use represents a complex mosaic of crop land, pasture and forest. Therefore, an inverse relationship between FC and IC may not be universal for subwatersheds that have witnessed many cycles of deforestation and cultivation.

It should come as little surprise that the progressive loss of FC has been linked to declining stream quality indicators, given that forested watersheds are often routinely used to define natural reference conditions for streams (Booth, 2000 and Horner *et al.*, 2001). Mature forest is considered to be the main benchmark for defining pre-development hydrology within a subwatershed, as well. Consequently, FC is perhaps the most powerful indicator to predict the quality of streams within the "sensitive" category (zero to 10% IC).

To use an extreme example, one would expect that stream quality indicators would respond quite differently in a subwatershed that had 90% FC compared to one that had 90% crop cover. Indeed, Booth (1991) suggests that stream quality can only be maintained when IC is limited to less than 10% <u>and</u> at least 65% FC is retained within a subwatershed. The key management implication then is that stream health is best managed by simultaneously minimizing the creation of IC and maximizing the preservation of native FC.

FC has also been shown to be useful in predicting the quality of terrestrial variables in a subwatershed. For example, the Mid-Atlantic Integrated Assessment (USEPA, 2000) has documented that watershed FC can reliably predict the diversity of bird, reptile and amphibian communities in the mid-Atlantic region. Moreover, the emerging discipline of landscape ecology provides watershed managers with a strong scientific foundation for deciding where FC should be conserved in a watershed. Conservation plans that protect and connect large forest fragments have been shown to be effective in conserving terrestrial species.

Riparian forest continuity has also shown considerable promise in predicting at least some indicators of stream quality for urban watersheds. Researchers have yet to come up with a standard definition of riparian continuity, but it is usually defined as the proportion of the perennial stream network in a subwatershed that has a fixed width of mature streamside forest. A series of studies indicates that aquatic insect and fish diversity are associated with high levels of riparian continuity (Horner *et al.*, 2001; May *et al.*, 1997; MNCPPC, 2000; Roth *et al.*, 1998). On the other hand, not much evidence has been presented to support the notion that riparian continuity has a strong influence on hydrology or water quality indicators.

One watershed variable that received little attention is the fraction of watershed area maintained in turf cover (TC). Grass often comprises the largest fraction of land area within low-density residential development and could play a significant role in streams that fall within the "impacted" category (10 to 25% IC). Although lawns are pervious, they have sharply different properties than the forests and farmlands they replace (i.e., irrigation, compacted soils, greater runoff, and much higher input of fertilizers and pesticides, etc.). It is interesting to speculate whether the combined area of IC and TC might provide better predictions about stream health than IC area alone, particularly within impacted subwatersheds.

Several other watershed variables might have at least supplemental value in predicting stream quality. They include the presence of extensive wetlands and/or beaverdam complexes in a subwatershed; the dominant form of drainage present in the watershed (tile drains, ditches, swales, curb and gutters, storm drain pipes); the average age of development; and the proximity of sewer lines to the stream. As far as we could discover, none of these variables has been systematically tested in a controlled population of small watersheds. We have observed that these factors could be important in our field investigations and often measure them to provide greater insight into subwatershed behavior.

Lastly, several watershed variables that are closely related to IC have been proposed to predict stream quality. These include population, percent urban land, housing density, road density and other indices of watershed development. As might be expected, they generally track the same trend as IC, but each has some significant technical limitations and/or difficulties in actual planning applications (Brown, 2000).

Individual vs. Multiple Indicators

The ICM does not predict the precise score of individual stream quality indicators, but rather predicts the average behavior of a group of indicators over a range of IC. Extreme care should be exercised if the ICM is used to predict the fate of individual indicators and/or species. This is particularly true for sensitive aquatic species, such as trout, salmon, and freshwater mussels. When researchers have examined the relationship between IC and individual species, they have often discovered lower thresholds for harm. For example, Boward *et al.* (1999) found that brook trout were not found in subwatersheds that had more than 4% IC in Maryland, whereas Horner and May (1999) asserted an 8% threshold for sustaining salmon in Puget Sound streams.

The key point is that if watershed managers want to maintain an individual species, they should be very cautious about adopting the 10% IC threshold. The essential habitat requirements for many sensitive or endangered species are probably determined by the *most sensitive* stream quality indicators, rather than the *average behavior* of all stream quality indicators.

Direct Causality vs. Association

A strong relationship between IC and declining stream quality indicators does not always mean that the IC is directly responsible for the decline. In some cases, however, causality can be demonstrated. For example, increased stormwater runoff volumes are directly caused by the percentage of IC in a subwatershed, although other factors such as conveyance, slope and soils may play a role.

In other cases, the link is much more indirect. For these indicators, IC is merely an index of the cumulative amount of watershed development, and more IC simply means that a greater number of known or unknown pollutant sources or stressors are present. In yet other cases, a causal link appears likely but has not yet been scientifically demonstrated. A good example is the more than 50 studies that have explored how fish or aquatic insect diversity changes in response to IC. While the majority of these studies consistently shows a very strong negative association between IC and biodiversity, they do not really establish which stressor or combination of stressors contributes most to the decline. The widely accepted theory is that IC changes stream hydrology, which degrades stream habitat, and in turn leads to reduced stream biodiversity.

Regional Differences

Currently, the ICM has been largely confirmed within the following regions of North America: the mid-Atlantic, the Northeast, the Southeast, the upper Midwest and the Pacific Northwest. Limited testing in Northern California, the lower Midwest and Central Texas generally agrees with the ICM. The ICM has not been tested in Florida, the Rocky Mountain West, and the Southwest. For a number of reasons, it is not certain if the ICM accurately predicts biological indicators in arid and semiarid climates (Maxted, 1999).

Measuring Impervious Cover

Most researchers have relied on total impervious cover as the basic unit to measure IC at the subwatershed level. The case has repeatedly been made that effective impervious cover is probably a superior metric (e.g., only counting IC that is hydraulically connected to the drainage system). Notwithstanding, most researchers have continued to measure total IC because it is generally quicker and does not require extensive (and often subjective) engineering judgement as to whether it is connected or not. Researchers have used a wide variety of techniques to estimate subwatershed IC, including satellite imagery, analysis of aerial photographs, and derivation from GIS land use layers. Table 2 presents some standard land use/IC relationships that were developed for suburban regions of the Chesapeake Bay.

Table 2: Land Use/IC Relationships for Suburban Areas of the Chesapeake Bay
(Cappiella and Brown, 2001)

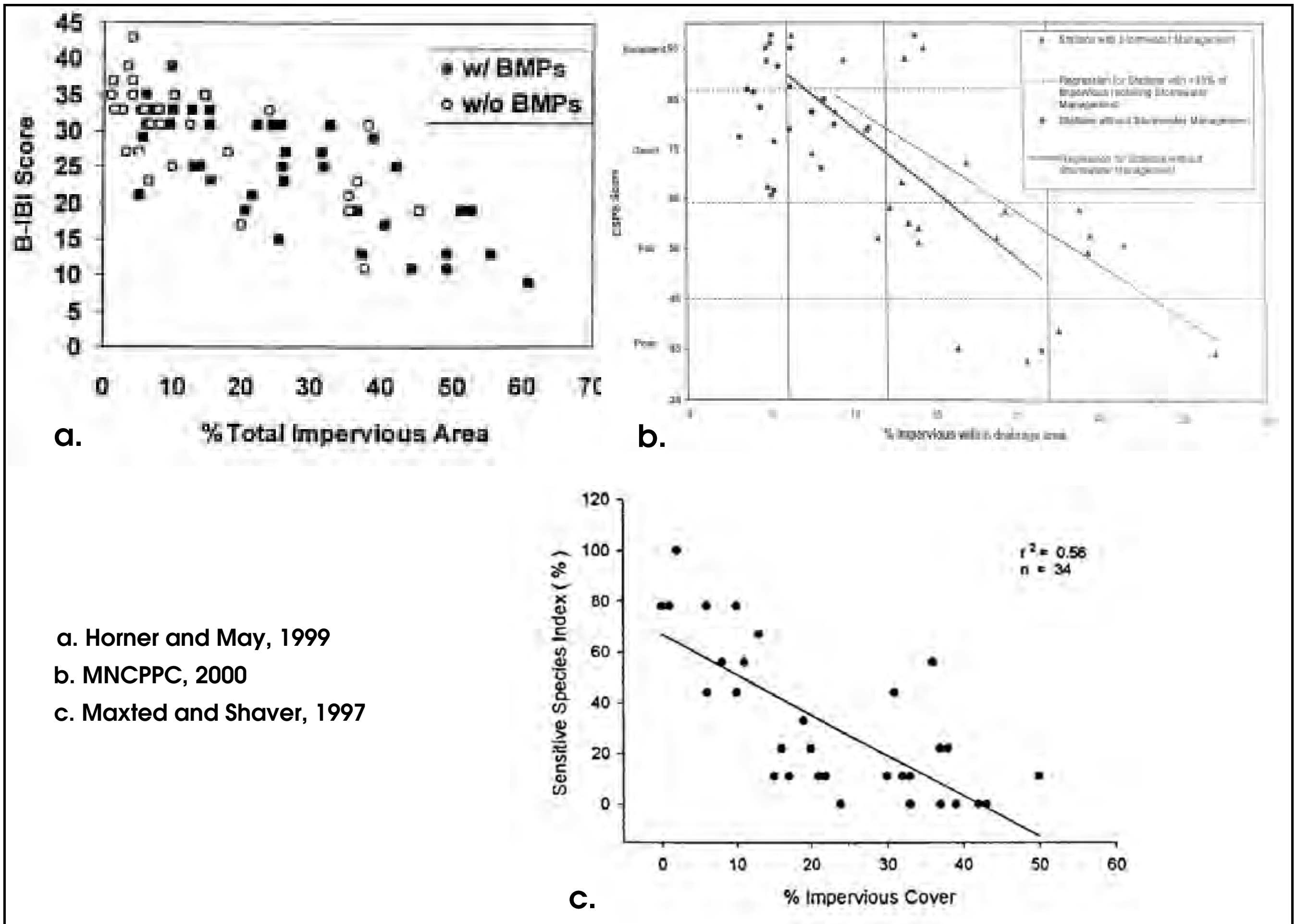
Land Use Category	Sample Number (N)	Mean IC (SE)	Land Use Category	Sample Number (N)	Mean IC (SE)
Agriculture	8	**1.9** – 0.3	Institutional	30	**34.4** – 3.45
Open Urban Land	11	**8.6** – 1.64	Light	20	**53.4** – 2.8
2 Acre Lot Residential	12	**10.6** – 0.65	Commercia	23	**72.2** – 2.0
1 Acre Lot Residential	23	**14.3** – 0.53	Churches	8	**39.9** – 7.8 1
1/2 Acre Lot Residential	20	**21.2** – 0.78	Schools	13	**30.3** – 4.8
1/4 Acre Lot Residential	23	**27.8** – 0.60	Municipals	9	**35.4** – 6.3
1/8 Acre Lot Residential	10	**32.6** – 1.6	Golf	4	**5.0** – 1.7
Townhome Residential	20	**40.9** – 1.39	Cemeteries	3	**8.3** – 3.5
Multifamily Residential	18	**44.4** – 2.0	Parks	4	**12.5** – 0.7

Three points are worth noting. First, it is fair to say that most researchers have spent more quality control effort on their stream quality indicator measurements than on their subwatershed IC estimates. At the current time, no standard protocol exists to estimate subwatershed IC, although Cappiella and Brown (2001) presented a useful method. At best, the different methods used to measure IC make it difficult to compare results from different studies, and at worst, it can introduce an error term of perhaps +/- 10% from the true value within an individual subwatershed. Second, it is important to keep in mind that IC is not constant over time; indeed, major changes in subwatershed IC have been observed within as few as two years. Consequently, it is sound practice to obtain subwatershed IC estimates from the most recent possible mapping data, to ensure that it coincides with stream quality indicator measurements. Lastly, it is important to keep in mind that most suburban and even rural zoning categories exceed 10% IC (see Table 2). Therefore, from a management standpoint, planners should try to project future IC, in order to determine the future stream classification for individual subwatersheds.

1.1.3 Influence of Watershed Treatment Practices on the ICM

The most hotly debated question about the ICM is whether widespread application of watershed practices such as stream buffers or stormwater management can mitigate the impact of IC, thereby allowing greater development density for a given watershed. At this point in time, there are fewer than 10 studies that directly bear on this critical question. Before these are reviewed, it is instructive to look at the difficult technical and scientific issues involved in detecting the effect of watershed treatment, given its enormous implications for land use control and watershed management.

The first tough issue is how to detect the effect of watershed treatment, given the inherent scatter seen in the IC/stream quality indicator relationship. Figure 4 illustrates the “double scatter” problem, based on three different urban stream research studies in Delaware, Maryland and Washington. A quick inspection of the three plots shows how intrinsically hard it is to distinguish the watershed treatment effect. As can be seen, stream quality indicators in subwatersheds with treatment tend to



a. Horner and May, 1999
b. MNCPPC, 2000
c. Maxted and Shaver, 1997

Figure 4: The Double Scatter Problem: Difficulties in Detecting the Effect of Watershed Treatment

overplot those in subwatersheds that lack treatment. While subtle statistical differences may be detected, they are not visibly evident. This suggests that the impact of watershed treatment would need to be extremely dramatic to be detected, given the inherent statistical variability seen in small watersheds (particularly so within the five to 25% IC range where scatter is considerable).

In an ideal world, a watershed study design would look at a controlled population of small urban watersheds that were developed with and without watershed practices to detect the impact of "treatment." In the real world, however, it is impossible to strictly control subwatershed variables. Quite simply, no two subwatersheds are ever alike. Each differs slightly with respect to drainage area, IC, forest cover, riparian continuity, historical land use, and percent watershed treatment. Researchers must also confront other real world issues when designing their watershed treatment experiments.

For example, researchers must carefully choose which indicator or group of indicators will be used to define stream health. IC has a negative influence on 26 stream quality indicators, yet nearly all of the watershed treatment research so far has focused on just a few biological indicators (e.g., aquatic insect or fish diversity) to define stream health. It is conceivable that watershed treatment might have no effect on biological indicators, yet have a positive influence on hydrology, habitat or water quality indicators. At this point, few of these indicators have been systematically tested in the field. It is extremely doubtful that any watershed practice can simultaneously improve or mitigate all 26 stream quality indicators, so researchers must carefully interpret the outcomes of their watershed treatment experiments.

The second issue involves how to quantify watershed treatment. In reality, watershed treatment collectively refers to dozens of practices that are installed at individual development sites in the many years or even decades it takes to fully "build out" a subwatershed. Several researchers have discovered that watershed practices are seldom installed consistently across an entire subwatershed. In some cases, less than a third of the IC in a subwatershed was actually treated by any practice, because development occurred prior to regulations; recent projects were exempted, waived or grandfathered; or practices were inadequately constructed or maintained (Horner and May, 1999 and MNCPPC, 2000).

Even when good coverage is achieved in a watershed, such as the 65 to 90% reported in studies of stormwater ponds (Jones *et al*., 1996; Maxted, 1999; Maxted and Shaver, 1997), it is still quite difficult to quantify the actual quality of treatment. Often, each subwatershed contains its own unique mix of stormwater practices installed over several decades, designed under diverse design criteria, and utilizing widely different stormwater technologies. Given these inconsistencies, researchers will need to develop standard protocols to define the extent and quality of watershed treatment.

Effect of Stormwater Ponds

With this in mind, the effect of stormwater ponds and stream buffers can be discussed. The effect of larger stormwater ponds in mitigating the impacts of IC in small watersheds has received the most scrutiny to date. This is not surprising, since larger ponds often control a large fraction of their contributing subwatershed area (e.g. 100 to 1,000 acres) and are located on the stream itself, therefore lending themselves to easier monitoring. Three studies have evaluated the impact of large stormwater ponds on downstream aquatic insect communities (Jones *et al.*, 1996; Maxted and Shaver, 1997; Stribling *et al.*, 2001). Each of these studies was conducted in small headwater subwatersheds in the mid-Atlantic Region, and none was able to detect major differences in aquatic insect diversity in streams with or without stormwater ponds.

Four additional studies statistically evaluated the stormwater treatment effect in larger populations of small watersheds with varying degrees of IC (Horner and May, 1999; Horner *et al.*, 2001; Maxted, 1999; MNCPPC, 2000). These studies generally sampled larger watersheds that had many stormwater practices but not necessarily complete watershed coverage. In general, these studies detected a small but positive effect of stormwater treatment relative to aquatic insect diversity. This positive effect was typically seen only in the range of five to 20% IC and was generally undetected beyond about 30% IC. Although each author was hesitant about interpreting his results, all generally agreed that perhaps as much as 5% IC could be added to a subwatershed while maintaining aquatic insect diversity, given effective stormwater treatment. Forest retention and stream buffers were found to be very important, as well. Horner *et al.* (2001) reported a somewhat stronger IC threshold for various species of salmon in Puget Sound streams.

Some might conclude from these initial findings that stormwater ponds have little or no value in maintaining biological diversity in small streams. However, such a conclusion may be premature for several reasons. First, the generation of stormwater ponds that was tested was not explicitly designed to protect stream habitat or to prevent downstream channel erosion, which would presumably promote aquatic diversity. Several states have recently changed their stormwater criteria to require extended detention for the express purpose of preventing downstream channel erosion, and these new criteria may exert a stronger influence on aquatic diversity. Instead, their basic design objective was to maximize pollutant removal, which they did reasonably well.

The second point to stress is that streams with larger stormwater ponds should be considered “regulated streams” (Ward and Stanford, 1979), which have a significantly altered aquatic insect community downstream of the ponds. For example, Galli (1988) has reported that on-stream wet stormwater ponds shift the trophic structure of the aquatic insect community. The insect community above the pond was dominated by shredders, while the insect community below the pond was dominated by scrapers, filterers and collectors. Of particular note, several pollution-sensitive species were eliminated below the pond. Galli reported that changes in stream temperatures, carbon supply and substrate fouling were responsible for the downstream shift in the aquatic insect community. Thus, while it is clear that large stormwater ponds can be expected to have a negative effect on aquatic insect diversity, they could still exert positive influence on other stream quality indicators.

Effect of Stream Buffers

A handful of studies have evaluated biological indicator scores for urban streams that have extensive forest buffers, compared to streams where they were mostly or completely absent (Horner and May, 1999; Horner *et al.*, 2001; May *et al.*, 1997; MNCPPC, 2000; Roth *et al.*, 1998; Steedman, 1988). Biological indicators included various indices of aquatic insect, fish and salmon diversity. Each study sampled a large population of small subwatersheds over a range of IC and derived a quantitative measure to express the continuity, width and forest cover of the riparian buffer network within each subwatershed. Riparian forests were hypothesized to have a positive influence on stream biodiversity, given the direct ways they contribute to stream habitat (e.g., shading, woody debris, leaf litter, bank stability, and organic carbon supply).

All five studies detected a small to moderate positive effect when forested stream buffers were present (frequently defined as at least two-thirds of the stream network with at least 100 feet of stream side forest). The greatest effect was reported by Horner and May (1999) and Horner *et al.* (2001) for salmon streams in the Puget Sound ecoregion. If excellent riparian habitats were preserved, they generally reported that fish diversity could be maintained up to 15% IC, and good aquatic insect diversity could be maintained with as much as 30% IC. Steedman (1988) reported a somewhat smaller effect for Ontario streams. MNCPPC (2000), May *et al.* (1997), and Roth *et al.* (1998) could not find a statistically significant relationship between riparian quality and urban stream quality indicators but did report that most outliers (defined as higher IC subwatersheds with unusually high biological indicator scores) were generally associated with extensive stream side forest.

1.1.4 Recommendations for Further ICM Research

At this point, we recommend three research directions to improve the utility of the ICM for watershed managers. The **first direction** is to expand basic research on the relationship between IC and stream quality indicators that have received little scrutiny. In particular, more work is needed to define the relationship between IC and hydrological and physical indicators such as the following:

- Physical loss or alteration of the stream network
- Stream habitat measures
- Riparian continuity
- Baseflow conditions during dry weather

In addition, more watershed research is needed in ecoregions and physiographic areas where the ICM has not yet been widely tested. Key areas include Florida, arid and semiarid climates, karst areas and mountainous regions. The basic multiple subwatershed monitoring protocol set forth by Schueler (1994a) can be used to investigate IC/stream quality relationships, although it would be wise to measure a wider suite of subwatershed variables beyond IC (e.g., forest cover, turf cover, and riparian continuity).

The **second** research direction is to more clearly define the impact of watershed treatment on stream quality indicators. Based on the insurmountable problems encountered in controlling variation at the subwatershed level, it may be necessary to abandon the multiple watershed or paired watershed sampling approaches that have been used to date. Instead, longitudinal monitoring studies within individual subwatersheds may be a more powerful tool to detect the effect of watershed treatment. These studies could track changes in stream quality indicators in individual subwatersheds over the entire development cycle: pre-development land use, clearing, construction, build out, and post construction. In most cases, longitudinal studies would take five to 10 years to complete, but they would allow watershed managers to measure and control the inherent variability at the subwatershed level and provide a “before and after” test of watershed treatment. Of course, a large population of test subwatersheds would be needed to satisfactorily answer the watershed treatment question.

The **third** research direction is to monitor more non-supporting streams, in order to provide a stronger technical foundation for crafting more realistic urban stream standards and to see how they respond to various watershed restoration treatments. As a general rule, most researchers have been more interested in the behavior of sensitive and impacted streams. The non-supporting stream category spans a wide range of IC, yet we do not really understand how stream quality indicators behave over the entire 25 to 100% IC range.

For example, it would be helpful to establish the IC level at the upper end of the range where streams are essentially transformed into an artificial conveyance system (i.e., become pipes or artificial channels). It would also be interesting to sample more streams near the lower end of the non-supporting category (25 to 35% IC) to detect whether stream quality indicators respond to past watershed treatment or current watershed restoration efforts. For practical reasons, the multiple subwatershed sampling approach is still recommended to characterize indicators in non-supporting streams. However, researchers will need to screen a large number of non-supporting subwatersheds in order to identify a few subwatersheds that are adequate for subsequent sampling (i.e., to control for area, IC, development age, percent watershed treatment, type of conveyance systems, etc.).

1.2 Impacts of Urbanization on Downstream Receiving Waters

In this section, we review the impacts of urbanization on downstream receiving waters, primarily from the standpoint of impacts caused by poor stormwater quality. We begin by looking at the relationship between IC and stormwater pollutant loadings. Next, we discuss the sensitivity of selected downstream receiving waters to stormwater pollutant loads. Lastly, we examine the effect of watershed treatment in reducing stormwater pollutant loads.

1.2.1 Relationship Between Impervious Cover and Stormwater Quality

Urban stormwater runoff contains a wide range of pollutants that can degrade downstream water quality (Table 3). Several generalizations can be supported by the majority of research conducted to date. First, the unit area pollutant load delivered by stormwater runoff to receiving waters increases in direct proportion to watershed IC. This is not altogether surprising, since pollutant load is the product of the average pollutant concentration and stormwater runoff volume. Given that runoff volume increases in direct proportion to IC, pollutant loads must automatically increase when IC increases, as long the average pollutant concentration stays the same (or increases). This relationship is a central assumption in most simple and complex pollutant loading models (Bicknell *et al.*, 1993; Donigian and Huber, 1991; Haith *et al.*, 1992; Novotny and Chester, 1981; NVPDC, 1987; Pitt and Voorhees, 1989).

The second generalization is that stormwater pollutant concentrations are generally similar

Table 3: Summary of Urban Stormwater Pollutant Loads on Quality of Receiving Waters

Pollutants in Urban Stormwater	WQ Impacts To:					Higher Unit Load?	Load a function of IC?	Other Factors Important in Loading
	R	L	E	A	W			
Suspended Sediment	Y	Y	Y	N	Y	Y (ag)	Y	channel erosion
Total Nitrogen	N	N	Y	Y	N	Y (ag)	Y	septic systems
Total Phosphorus	Y	Y	N	N	Y	Y (ag)	Y	tree canopy
Metals	Y	Y	Y	?	N	Y	Y	vehicles
Hydrocarbons	Y	Y	Y	Y	Y	Y	?	related to VMTs and hotspots
Bacteria/Pathogens	Y	Y	Y	N	Y	Y	Y	many sources
Organic Carbon	N	?	?	?	Y	Y	Y	
MTBE	N	N	N	Y	Y	Y	?	roadway, VMTs
Pesticides	?	?	?	?	Y	Y	?	turf/landscaping
Chloride	?	Y	N	Y	Y	Y	?	road density
Trash/Debris	Y	Y	Y	N	?	Y	Y	curb and gutters

Major Water Quality Impacts Reported for:
R = River, L = Lake, E = Estuary, A = Aquifer, W = Surface Water Supply
Higher Unit Area Load? Yes (compared to all land uses) (ag): with exception of cropland
Load a function of IC? Yes, increases proportionally with IC

at the catchment level, regardless of the mix of IC types monitored (e.g., residential, commercial, industrial or highway runoff). Several hundred studies have examined stormwater pollutant concentrations from small urban catchments and have generally found that the variation within a catchment is as great as the variation between catchments. Runoff concentrations tend to be log-normally distributed, and therefore the long term "average" concentration is best expressed by a median value. It should be kept in mind that researchers have discovered sharp differences in pollutant concentrations for smaller, individual components of IC (e.g., rooftops, parking lots, streets, driveways and the like). Since most urban catchments are composed of many kinds of IC, this mosaic quality tempers the variability in long term pollutant concentrations at the catchment or subwatershed scale.

The third generalization is that median concentrations of pollutants in urban runoff are usually higher than in stormwater runoff from most other non-urban land uses. Consequently, the unit area nonpoint pollutant load generated by urban land normally exceeds that of nearly all watershed land uses that it replaces (forest, pasture, cropland, open space — see Table 3). One important exception is cropland, which often produces high unit area sediment and nutrient loads in many regions of the country. In these watersheds, conversion of intensively managed crops to low density residential development may actually result in a slightly decreased sediment or nutrient load. On the other hand, more intensive land development (30% IC or more) will tend to equal or exceed cropland loadings.

The last generalization is that the effect of IC on stormwater pollutant loadings tends to be weakest for subwatersheds in the one to 10% IC range. Numerous studies have suggested that other watershed and regional factors may have a stronger influence, such as the underlying geology, the amount of carbonate rock in the watershed, physiographic region, local soil types, and most important, the relative fraction of forest and crop cover in the subwatershed (Herlihy *et al*., 1998 and Liu *et al.,* 2000). The limited influence of IC on pollutant loads is generally consistent with the finding for hydrologic, habitat and biological indicators over this narrow range of IC. Once again, watershed managers are advised to track other watershed indicators in the sensitive stream category, such as forest or crop cover.

1.2.2 Water Quality Response to Stormwater Pollution

As noted in the previous section, most ICM research has been done on streams, which are directly influenced by increased stormwater. Many managers have wondered whether the ICM also applies to downstream receiving waters, such as lakes, water supply reservoirs and small estuaries. In general, the exact water quality response of downstream receiving waters to increased nonpoint source pollutant loads depends on many factors, including the specific pollutant, the existing loading generated by the converted land use, and the geometry and hydraulics of the receiving water. Table 3 indicates the sensitivity of rivers, lakes, estuaries, aquifers and water supply reservoirs to various stormwater pollutants.

Lakes and the ICM

The water column and sediments of urban lakes are impacted by many stormwater pollutants, including sediment, nutrients, bacteria, metals, hydrocarbons, chlorides, and trash/debris. Of these pollutants, limnologists have always regarded phosphorus as the primary lake management concern, given that more than 80% of urban lakes experience symptoms of eutrophication (CWP, 2001a).

In general, phosphorus export steadily increases as IC is added to a lake watershed, although the precise amount of IC that triggers eutrophication problems is unique to each urban lake. With a little effort, it is possible to calculate the specific IC threshold for an individual lake, given its internal geometry, the size of its contributing watershed, current in-lake phosphorus concentration, degree of watershed treatment, and the desired water quality goals for the lake (CWP, 2001a). As a general rule, most lakes are extremely sensitive to increases in phosphorus loads caused by watershed IC. Exceptions include lakes that are unusually deep and/or have very small drainage area/lake area ratios. In most lakes, however, even a small amount of watershed development will result in an upward shift in trophic status (CWP, 2001a).

Reservoirs and the ICM

While surface water supply reservoirs respond to stormwater pollutant loads in the same general manner as lakes, they are subject to stricter standards because of their uses for drinking water. In particular, water supply reservoirs are particularly sensitive to increased turbidity, pathogens, total organic carbon, chlorides, metals, pesticides and hydrocarbon loads, in addition to phosphorus (Kitchell, 2001). While some pollutants can be removed or reduced through expanded filtering and treatment at drinking water intakes, the most reliable approach is to protect the source waters through watershed protection and treatment.

Consequently, we often recommend that the ICM be used as a "threat index" for most drinking water supplies. Quite simply, if current or future development is expected to exceed 10% IC in the contributing watershed, we recommend that a very aggressive watershed protection strategy be implemented (Kitchell, 2001). In addition, we contend that drinking water quality cannot be sustained once watershed IC exceeds 25% and have yet to find an actual watershed where a drinking water utility has been maintained under these conditions.

Small Tidal Estuaries and Coves and the ICM

The aquatic resources of small tidal estuaries, creeks, and coves are often highly impacted by watershed development and associated activities, such as boating/marinas, wastewater discharge, septic systems, alterations in freshwater flow and wetland degradation and loss. Given the unique impacts of eutrophication on the marine system and stringent water quality standards for shellfish harvesting, the stormwater pollutants of greatest concern in the estuarine water column are nitrogen and fecal coliform bacteria. Metals and hydrocarbons in stormwater runoff can also contaminate bottom sediments, which can prove toxic to local biota (Fortner *et al*., 1996; Fulton *et al.*, 1996; Kucklick *et al.*, 1997; Lerberg *et al.*, 2000; Sanger *et al.*, 1999; Vernberg *et al.*, 1992).

While numerous studies have demonstrated that physical, hydrologic, water quality and biological indicators differ in urban and non-urban coastal watersheds, only a handful of studies have used watershed IC as an indicator of estuarine health. These studies show significant correlations with IC, although degradation thresholds may not necessarily adhere to the ICM due to tidal dilution and dispersion. Given the limited research, it is not fully clear if the ICM can be applied to coastal systems without modification.

Atmospheric deposition is considered a primary source of nitrogen loading to estuarine watersheds. Consequently, nitrogen loads in urban stormwater are often directly linked to IC. Total nitrogen loads have also been linked to groundwater input, especially from subsurface discharges from septic systems, which are common in low density coastal development (Swann, 2001; Valiela *et al*., 1997; Vernberg *et al.*, 1996a). Nitrogen is generally considered to be the limiting nutrient in estuarine systems, and increased loading has been shown to increase algal and phytoplankton biomass and cause shifts in the phytoplankton community and food web structure that may increase the potential for phytoplankton blooms and fish kills (Bowen and Valiela, 2001; Evgenidou *et al.*, 1997; Livingston, 1996).

Increased nitrogen loads have been linked to declining seagrass communities, finfish populations, zooplankton reproduction, invertebrate species richness, and shellfish populations (Bowen and Valiela, 2001; Rutkowski *et al.*, 1999; Short and Wyllie-Echeverria, 1996; Valiela and Costa, 1988). Multiple studies have shown significant increases in nitrogen loading as watershed land use becomes more urban (Valiela *et al.*, 1997; Vernberg *et al.* 1996a; Wahl *et al.*, 1997). While a few studies link nitrogen loads with building and population density, no study was found that used IC as an indicator of estuarine nitrogen loading.

The second key water quality concern in small estuaries is high fecal coliform levels in stormwater runoff, which can lead to the closure of shellfish beds and swimming beaches. Waterfowl and other wildlife have also been shown to contribute to fecal coliform loading (Wieskel *et al.*, 1996). Recent research has shown that fecal coliform standards are routinely violated during storm events at very low levels of IC in coastal watersheds (Mallin *et al.*, 2001; Vernberg *et al.*, 1996b; Schueler, 1999). Maiolo and Tschetter (1981) found a significant correlation between human population and closed shellfish acreage in North Carolina, and Duda and Cromartie (1982) found greater fecal coliform densities when septic tank density and IC increased, with an approximate threshold at 10% watershed IC.

Recently, Mallin *et al.* (2000) studied five small North Carolina estuaries of different land uses and showed that fecal coliform levels were significantly correlated with watershed population, developed land and IC. Percent IC was the most statistically significant indicator and could explain 95% of the variability in fecal coliform concentrations. They also found that shellfish bed closures were possible in watersheds with less than 10% IC, common in watersheds above 10% IC, and almost certain in watersheds above 20% IC. While higher fecal coliform levels were observed in developed watersheds, salinity, flushing and proximity to pollution sources often resulted in higher concentrations at upstream locations and at high tides (Mallin *et al.*, 1999). While these studies support the ICM, more research is needed to prove the reliability of the ICM in predicting shellfish bed closures based on IC.

Several studies have also investigated the impacts of urbanization on estuarine fish, macrobenthos and shellfish communities. Increased PAH accumulation in oysters, negative effects of growth in juvenile sheepshead minnows, reduced molting efficiency in copepods, and reduced numbers of grass shrimp have all been reported for urban estuaries as compared to forested estuaries (Fulton *et al.*, 1996). Holland *et al. (*1997) reported that the greatest abundance of penaid shrimp and mummichogs was observed in tidal creeks with forested watersheds compared to those with urban cover. Porter *et al.* (1997) found lower grass shrimp abundance in small tidal creeks adjacent to commercial and urban development, as compared to non-urban watersheds.

Lerberg *et al.* (2000) studied small tidal creeks and found that highly urban watersheds (50% IC) had the lowest benthic diversity and abundance as compared to suburban and forested creeks, and benthic communities were numerically dominated by tolerant oligochaetes and polychaetes. Suburban watersheds (15 to 35% IC) also showed signs of degradation and had some pollution tolerant macrobenthos, though not as markedly as urban creeks. Percent abundance of pollution-indicative species showed a marked decline at 30% IC, and the abundance of pollution-sensitive species also significantly correlated with IC (Lerberg *et al*., 2000). Holland *et al*. (1997) reported that the variety and food availability for juvenile fish species was impacted at 15 to 20% IC.

Lastly, a limited amount of research has focused on the direct impact of stormwater runoff on salinity and hypoxia in small tidal creeks. Blood and Smith (1996) compared urban and forested watersheds and found higher salinities in urban watersheds due to the increased number of impoundments. Fluctuations in salinity have been shown to affect shellfish and other aquatic populations (see Vernberg, 1996b). When urban and forested watersheds were compared, Lerberg *et al.* (2000) reported that higher salinity fluctuations occurred most often in developed watersheds; significant correlations with salinity range and IC were also determined. Lerberg *et al.* (2000) also found that the most severe and frequent hypoxia occurred in impacted salt marsh creeks and that dissolved oxygen dynamics in tidal creeks were comparable to dead-end canals common in residential marina-style coastal developments. Suburban watersheds (15 to 35% IC) exhibited signs of degradation and had some pollution-tolerant macrobenthic species, though not to the extent of urban watersheds (50% IC).

In summary, recent research suggests that indicators of coastal watershed health are linked to IC. However, more research is needed to clarify the relationship between IC and estuarine indicators in small tidal estuaries and high salinity creeks.

1.2.3 Effect of Watershed Treatment on Stormwater Quality

Over the past two decades, many communities have invested in watershed protection practices, such as stormwater treatment practices (STPs), stream buffers, and better site design, in order to reduce pollutant loads to receiving waters. In this section, we review the effect of watershed treatment on the quality of stormwater runoff.

Effect of Stormwater Treatment Practices
We cannot directly answer the question as to whether or not stormwater treatment practices can significantly reduce water quality impacts at the watershed level, simply because no controlled monitoring studies have yet been conducted at this scale. Instead, we must rely on more indirect research that has tracked the change in mass or concentration of pollutants as they travel through individual stormwater treatment practices. Thankfully, we have an abundance of these performance studies, with nearly 140 monitoring studies evaluating a diverse range of STPs, including ponds, wetlands, filters, and swales (Winer, 2000).

These studies have generally shown that stormwater practices have at least a moderate ability to remove many pollutants in urban stormwater. Table 4 provides average removal efficiency rates for a range of practices and stormwater pollutants, and Table 5 profiles the mean storm outflow concentrations for various practices. As can be seen, some groups of practices perform better than others in removing certain stormwater pollutants. Consequently, managers need to carefully choose which practices to apply to solve the primary water quality problems within their watersheds.

It is also important to keep in mind that site-based removal rates cannot be extrapolated to the watershed level without significant adjustment. Individual site practices are never implemented perfectly or consistently across a watershed. At least three discount factors need to be considered: bypassed load, treatability and loss of performance over time. For a review on how these discounts are derived, consult Schueler and Caraco (2001). Even under the most optimistic watershed implementation scenarios, overall pollutant reduc-

Table 4: The Effectiveness of Stormwater Treatment Practices in Removing Pollutants - Percent Removal Rate (Winer, 2000)

Practice	N	TSS	TP	OP	TN	NOx	Cu	Zn	Oil/ Grease[1]	Bacteria
Dry Ponds	9	47	19	N/R	25	3.5	26	26	3	44
Wet Ponds	43	80	51	65	33	43	57	66	78	70
Wetlands	36	76	49	48	30	67	40	44	85	78
Filtering Practices[2]	18	86	59	57	38	-14	49	88	84	37
Water Quality Swales	9	81	34	1.0	84	31	51	71	62	-25
Ditches[3]	9	31	-16	N/R	-9.0	24	14	0	N/R	0
Infiltration	6	95	80	85	51	82	N/R	N/R	N/R	N/R

1: Represents data for Oil and Grease and PAH
2: Excludes vertical sand filters
3: Refers to open channel practices not designed for water quality
N/R = Not Reported

Practice	N	TSS	TP	OP	TN	NOx	Cu[1]	Zn[1]
Dry Ponds[2]	3	28	0.18	N/R	0.86	N/R	9.0	98
Wet Ponds	25	17	0.11	0.03	1.3	0.26	5.0	30
Wetlands	19	22	0.20	0.07	1.7	0.36	7.0	31
Filtering Practices[3]	8	11	0.10	0.07	1.1	0.55	9.7	21
Water Quality Swales	7	14	0.19	0.09	1.1	0.35	10	53
Ditches[4]	3	29	0.31	N/R	2.4	0.72	18	32

Table 5: Median Effluent Concentrations from Stormwater Treatment Practices (mg/l) (Winer, 2000)

1. Units for Zn and Cu are micrograms per liter (Fg/l)
2. Data available for Dry Extended Detention Ponds only
3. Excludes vertical sand filters
4. Refers to open channel practices not designed for water quality
N/R = Not Reported

tions by STPs may need to be discounted by at least 30% to account for partial watershed treatment.

Even with discounting, however, it is evident that STPs can achieve enough pollutant reduction to mimic rural background loads for many pollutants, as long as the watershed IC does not exceed 30 to 35%. This capability is illustrated in Figure 5, which shows phosphorus load as a function of IC, with and without stormwater treatment.

Effect of Stream Buffers/Riparian Areas

Forested stream buffers are thought to have very limited capability to remove stormwater pollutants, although virtually no systematic monitoring data exists to test this hypothesis. The major reason cited for their limited removal capacity is that stormwater generated from upland IC has usually concentrated before it reaches the forest buffer and therefore crosses the buffer in a channel, ditch or storm drain pipe. Consequently, the opportunity to filter runoff is lost in many forest buffers in urban watersheds.

Effect of Better Site Design

Better site design (BSD) is a term for nonstructural practices that minimize IC, conserve natural areas and distribute stormwater treatment across individual development sites. BSD is also known by many other names, including conservation development, low-impact development, green infrastructure, and sustainable urban drainage systems. While

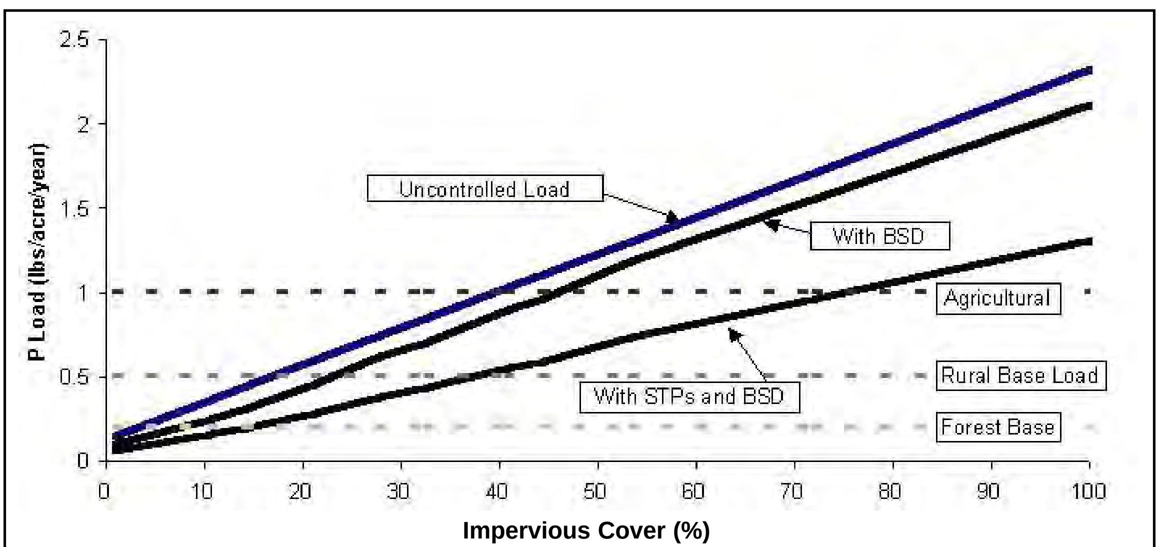


Figure 5: Estimated Phosphorus Load as a Function of Impervious Cover, Discounted Stormwater Treatment and Better Site Design (Schueler and Caraco, 2001)

some maintain that BSD is an alternative to traditional STPs, most consider it to be an important complement to reduce pollutant loads.

While BSD has become popular in recent years, only one controlled research study has evaluated its potential performance, and this is not yet complete (i.e. Jordan Cove, CT).

Indirect estimates of the potential value of BSD to reduce pollutant discharges have been inferred from modeling and redesign analyses (Zielinski, 2000). A typical example is provided in Figure 5, which shows the presumed impact of BSD in reducing phosphorus loadings. As is apparent, BSD appears to be a very effective strategy in the one to 25% IC range, but its benefits diminish beyond that point.